Available online at www.sciencedirect.com
SCIENCE DIRECT®
Aquacultural Engineering 34 (2006) 92–102
aquacultural engineering
www.elsevier.com/locate/aqua-online

Aerobic treatment of gravity thickening tank supernatant

Brian L. Brazil [a,*], Steven T. Summerfelt [b]

[a] US Department of Agriculture, Agricultural Research Service, National Center for Cool and Cold Water Aquaculture, 11876 Leetown Road, Kearneysville, WV 25430, USA
[b] The Conservation Fund Freshwater Institute, 1098 Turner Road, Shepherdstown, WV 25443, USA

Received 15 February 2005; accepted 13 June 2005

Abstract

Aerobic treatment of the supernatant overflowing an aquaculture manure-thickening tank was studied in replicated circular tank reactors (500 L) at hydraulic residence times (HRT) of 1, 3, and 6 days under cool (mean temperature = 13.5 °C) and warm (mean temperature = 19.3 °C) water conditions. Influent characteristics differed between temperatures, most likely reflecting changes in microbial activity occurring within the material contained in the gravity thickeners. Organic carbon and carbonaceous substances were the most readily removed during aerobic treatment. Soluble carbonaceous biological oxygen demand (cBOD) concentrations were decreased an average 91% across all HRTs at the warmer temperatures and 82% during the cool temperatures. Whether measured as soluble chemical oxygen demand (COD), or dissolved organic carbon (DOC), organic constituent removal efficiency ranged from 75 to 87% at all HRTs during both study phases. Total suspended solids (TSS) concentrations increased within the aerobic treatment vessels as soluble wastes were converted into heterotrophic and/or algae biomass. The increase in TSS concentration within the aerobic treatment vessel indicates that a solids capture process will be necessary to meet effluent suspended solids standards when employing the aerobic basin strategy. Total ammonia nitrogen (TAN) removal efficiency increased with increasing HRT, with an 87% removal efficiency achieved at the 6-day HRT under warm water conditions. During the cool temperature phase, the highest TAN reduction, 57%, was observed at the 3-day HRT. With respect to nitrite and nitrate concentration, effluent from the 1-day HRT treatment possessed the lowest concentrations under both temperature conditions. Dissolved phosphorus concentrations were reduced by an average 22% following treatment at the cool temperature. Under warm water conditions, phosphorus concentrations were reduced by 16.6, 42.6, and 64.7% for the 1-, 3-, and 6-day HRTs, respectively.

Keywords: Effluent treatment; Aerobic lagoon; Recirculating aquaculture; Solids thickening; Supernatant

1. Introduction

Effluent regulations for aquaculture facilities are becoming increasingly stringent at the state and local levels. In addition, EPA has developed national

* Corresponding author. Tel.: +1 304 724 8340x2135; fax: +1 304 725 0351.
E-mail address: bbrazil@ncccwa.ars.usda.gov (B.L. Brazil).

0144-8609/$ – see front matter © 2005 Elsevier B.V. All rights reserved.
doi:10.1016/j.aquaeng.2005.06.001

discharge standards focused on the implementation of Best Management Practices for the aquaculture industry (EPA, 2004). The objective of this action is to reduce nutrient loads to receiving waters. Most aquaculture facilities reduce their potential nutrient discharge through solids removal from their waste water (Bastian, 1992; IDEO, 1998).

Mechanical filters and/or gravitational separators are used to remove suspended and settleable solids from intensive aquaculture systems (Cripps and Kelly, 1996; Chen et al., 1997, 2002; Bergheim et al., 1998; Cripps and Bergheim, 2000; Ebeling et al., 2003, 2004; Davidson and Summerfelt, 2005). With microscreen drum filters, for example, the solids laden backwash flow produced is approximately 0.2–2.0% of the volume of the bulk flow filtered (Bergheim et al., 1998; Summerfelt, 1999). Whereas the solids stream produced from settling basins is periodic and associated with cleaning the basin. In such basins, the concentrated solids are often diluted by the water volume above the solids layer, reducing solids concentrations from between 3 and 5% to less than 500 mg/L.

The solids contained in the backwash from the mechanical filters or settling basins are often captured and stored in an off-line solids thickening basin until final disposal (Cripps and Kelly, 1996; Chen et al., 1997, 2002; Bergheim et al., 1998; Cripps and Bergheim, 2000; Ebeling et al., 2003, 2004; Adler and Sikora, 2005). Thickening in such vessels is used to reduce the volume of solids that require treatment. In doing so, more treatment options become viable as well as potentially reducing the transportation costs for additional processing or final disposal (Chen et al., 1997; Summerfelt et al., 1999; Cripps and Bergheim, 2000; Michael, 2003).

The organic solids tend to degrade and mineralize during storage (Cripps and Kelly, 1996; Summerfelt, 1999; Chen et al., 2002; Summerfelt and Vinci, 2003; Adler and Sikora, 2005). For example, monitoring conducted by personnel at the Conservation Fund Freshwater Institute revealed after 4 weeks of storage in a gravity thickening tank, supernatant (approximately 8 L/min) exiting the tanks possessed approximately 7–8 mg/L of total ammonia nitrogen (TAN), 6–10 mg/L of soluble phosphorus, and 80–140 mg/L of soluble carbonaceous biological oxygen demand (cBOD). These nutrient constituent concentrations, which resulted from biosolids degradation and nutrient leaching, were approximately 7-, 8.5-, and nearly 20-times higher than their respective concentrations in the backwash from the effluent drum filter. Therefore, this relatively small supernatant flow from the off-line gravity thickening tank will almost always contain the highest concentration of dissolved pollutants at a given recirculating facility (Summerfelt and Vinci, 2003). In fact, the waste concentrations found in this relatively small flow are more similar to the concentrations in wastewaters entering secondary treatment at publicly owned treatment works (POTWs) (Metcalf and Eddy, 2003). Therefore, many secondary treatment options used at POTWs can be used to treat thickening tank supernatant, as the daily volume produced is relatively small compared to the recirculating system processes flow, e.g., less than 0.2–2.0% of the recirculating system flow. If removal of particulate wastes and phosphorus are the goal, then the supernatant can be treated using coagulation–flocculation aids such as alum, ferric chloride, or polymers followed by settling or filtration (Ebeling et al., 2003). The tank overflow can also be reused for irrigation (Chen et al., 1997, 2002) or hydroponics (Adler et al., 1996, 2000). Removal of soluble cBOD and inorganic nitrogen compounds can be achieved with more traditional treatment processes, including: aerobic or anaerobic lagoons, created wetlands, anaerobic filters, or other suitable technologies. However, reducing effluent TAN levels, either through conversion to an oxidized form or algal assimilation, requires aerobic conditions, thereby eliminating most anaerobic treatment processes.

Lagoons have been successfully applied to reduce the nitrogen and phosphorus concentrations in the primary effluent streams from intensive aquaculture facilities (Chen et al., 1997, 2002). However, much of this work has focused on the efficacy of treating the full process flow (Bergheim and Brinker, 2003; Porrello et al., 2003a) and is often based on designs used for treating bovine and swine manure in lagoon treatment systems (Bastian, 1992). These basins tend to be oversized allowing for solids settling, accumulation and storage, and removal of nutrients, which are much higher in bovine and swine manures with respect to solids and cBOD than those observed in the primary waste stream from an aquaculture facility. While this design criteria is suitable for treating a concentrated

solids waste stream (IDEO, 1998), the land required to construct the treatment lagoon might not be available or too costly to acquire.

This study was conducted to characterize the impact of hydraulic residence time on aerobic treatment of supernatant from gravity thickening tanks. Additionally, the influence of temperature on treatment performance was examined by conducting the study during both summer and winter periods.

2. Materials and methods

2.1. Setup and operation

Nine replicated aeration tanks, simulating aerobic lagoons without the settling and sludge storage components, were set-up in a greenhouse at the Conservation Fund Freshwater Institute (Shepherdstown, West Virginia, USA). These tanks were used to treat the supernatant overflow from a solids thickening basin (Fig. 1). The radial-flow type solids thickening basin (Davidson and Summerfelt, 2005) was used to treat the water and solids slurry backwashed from a microscreen drum filter operated with 90-μm openings (Model RFM 4848, PRA Manufacturing Ltd., Nanaimo, British Columbia, Canada). Hydraulic retention times (HRT) of 1, 3 and 6 days were each tested in three tanks under summer (July–August, 2002 mean day length = 14 h) and winter (November–December 2002, mean day length = 10 h) months to evaluate the effect of seasonal conditions on treatment efficiency.

Each aerated tank (1.2 m diameter × 1 m deep) contained 500 L and operated as a completely mixed stirred reactor. Supernatant (influent material) from the gravity thickeners was collected in a polypropylene storage tank and then pumped at rates of 347, 116 and 58 mL/min for the 1-, 3- and 6-day HRT treatments, respectively. Peristaltic pumps, used to deliver influent, were checked weekly to ensure consistency and the delivery tubing cleaned three times weekly to eliminate solids build up and minimize influent flow fluctuations. Delivery tubing was cut to the same length for all nine tanks. Air supplied via two medium-pore ceramic stone diffusers was used for both aeration and suspension of particulate material.

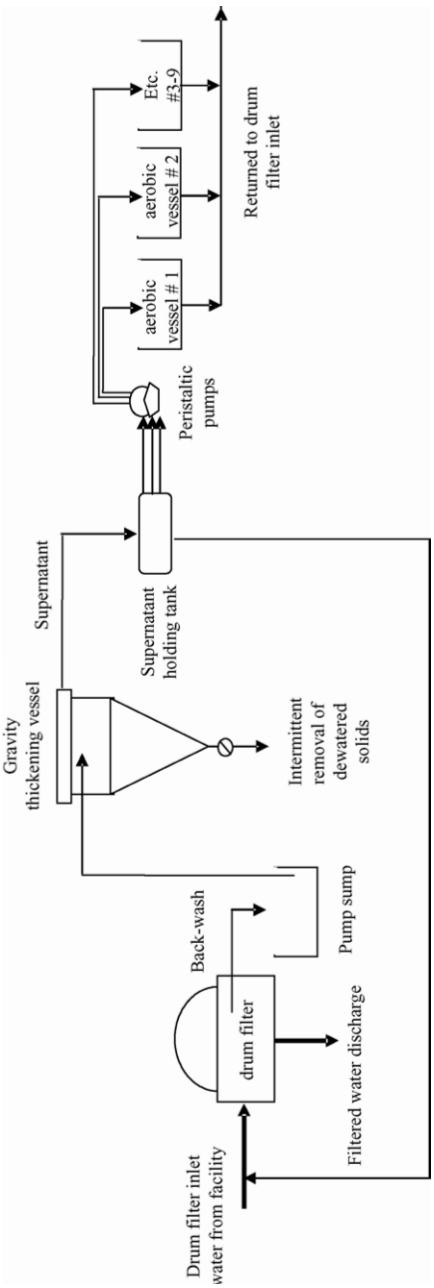

Fig. 1. Process flow schematic of the effluent treatment system employed at Freshwater Institute (Shepherdstown, WV). The thickening tank received and stored backwash from a microscreen drum filter. Supernatant overflowed to storage tank before being pumped back to the drum filter. During this study, the aerobic treatment tanks were set to treat a portion of the supernatant from the gravity thickening tank and this effluent discharged to a storage lagoon.

2.2. Data collection and analytical procedures

Tank water pH, temperature, and dissolved oxygen (DO) and sunlight intensity were measured daily. Influent and effluent samples were collected twice weekly for 6 weeks during both phases and were analyzed for concentrations of total suspended solids (TSS), volatile suspended solids (VSS), dissolved organic carbon (DOC), soluble cBOD, soluble chemical oxygen demand (sCOD), TAN, nitrite nitrogen (NO_2–N), nitrate nitrogen (NO_3–N), total and dissolved reactive phosphorus, (tPO_4–P and dPO_4–P, respectively). Influent was collected directly from the supernatant storage tank while effluent samples were collected from each tank reactor, under the assumption that the tanks were completely mixed. Immediately following collection, samples were placed in a cooler with ice packs and transported to the National Center for Cool and Cold Water Aquaculture (Kearneysville, WV) water quality laboratory for analysis.

A portable YSI-dissolved oxygen meter was used to measure DO and temperature (Yellow Springs, OH, USA) while pH was measured using an Accumet 915 pH meter (Fisher Scientific, Pittsburg, Pennsylvania, USA). Total suspended solids (TSS) and volatile suspended solids (VSS) were measured according to Standard Methods procedures 2540 D and 2540 E, respectively (APHA, 1998). Samples were filtered through a 1.2 μm glass microfiber filter (Whitman, Maidstone, Kent, UK) with the filtrate then used determine dissolved constituent concentrations. Daily solar radiation measurements were obtained from the USDA-ARS Appalachian Fruit Research Station (Kearneysville, West Virginia, USA). A quantum light sensor (Spectrum Technologies, Plainfield, Illinois, USA) measured and record photosynthetically active radiation (PAR), which is a visible light from 400 to 700 nm. Generally, 1800 μmol/m^2 s is considered full sun.

Dissolved organic carbon (DOC) was analyzed using a total organic carbon analyzer (Shimadzu model TOC-V, Kyoto, Japan) while chemical oxygen demand was measured according to Standard Methods procedure 5220 °C (APHA, 1998). During Phase 1, total ammonia nitrogen (TAN), nitrite (NO_2^-–N), and nitrate (NO_3^-–N) were analyzed using an Odyssey DR/2500 spectrophotometer following methods 8038, 8171 and 8507, respectively (HACH Company, Loveland, Colorado, USA).

During Phase 2, TAN, nitrite and nitrate concentrations were analyzed using ion chromatography (model IC 90, Dionex ICS-90, Sunnyvale, California, USA) with a conductivity detector. Prior to analysis, samples were filtered through a 0.45 μm Supor membrane filter. Periodically, results obtained from the IC were validated with colormetric analysis.

Reactive phosphorus, total and dissolved, was measured using an Odyssey DR/2000 spectrophotometer according to HACH method 8048 (HACH Company, Loveland, CO, USA). Sample aliquots used for tPO_4–P were only filtered through 1.2 μm glass microfarad filter, whereas aliquots analyzed for dPO_4 were filtered first through 1.2 μm glass fiber filter and then through 0.45 μm Supor membrane filters. Soluble cBOD was performed following Standard Methods procedure 5210 B 5-day tests (APHA, 1998) on samples filtered through 1.2 μm glass fiber filters.

The statistical analysis was performed using SAS statistical software (SAS Institute, Inc., Cary, North Carolina, USA). Treatment means were compared using two-way ANOVA where HRT and temperature were considered the main effects with Tukey's multiple comparison testing using to identify statistically different ($p < 0.05$ significance level) treatment group means.

3. Results and discussion

Physical water quality conditions observed during both phases of the study are presented in Table 1. DO concentrations within the treatment basins remained near saturation from the continuous aeration and photosynthetic processes (Hosetti and Frost, 1995). While concentrations were generally near saturation, daily and seasonal variations were observed (Fig. 2). Similar fluctuations are typically measured in pond environments and can be influenced by substrate and/or hydraulic loading, sunlight, and algal population dynamics (Porrello et al., 2003b). A Pearson's correlation evaluation did not identify any parameters beyond temperature (Pearson's correlation coefficient = 0.664, $p < 0.0001$) and HRT (Pearson's correlation coefficient = 0.621, $p < 0.0001$) that significantly contributed to the observed DO trends.

Table 1
Summary of physical–chemical water quality parameters from gravity thickener supernatant following aerobic treatment during summer and winter conditions

	Summer phase			Winter phase		
	1-day	3-day	6-day	1-day	3-day	6-day
Day light (h)		14			10	
Solar radiation/day (uM/m^2 s)		711 ± 231			301 ± 141	
Temperature (°C)						
Influent mean ± S.E.		19.3 ± 0.4^{A}			13.5 ± 0.1^{AB}	
Effluent mean ± S.E.	21.3 ± 0.40^{B}	22.2 ± 0.38^{C}	22.7 ± 0.40^{D}	13.5 ± 0.13^{A}	13.4 ± 0.15^{B}	13.4 ± 0.16^{B}
Dissolved oxygen (mg/l)						
Influent mean ± S.E.		1.76 ± 0.19^{A}			0.32 ± 0.04^{A}	
Effluent mean ± S.E.	6.95 ± 0.15^{B}	7.75 ± 0.16^{C}	8.26 ± 0.09^{D}	7.11 ± 0.16^{B}	9.32 ± 0.09^{C}	9.88 ± 0.08^{D}
pH (standard units)						
Influent mean ± S.E.		7.08 ± 0.07^{A}			7.13 ± 0.09^{A}	
Effluent mean ± SE	8.14 ± 0.03^{B}	8.39 ± 0.03^{C}	8.61 ± 0.04^{D}	7.96 ± 0.03^{B}	8.20 ± 0.02^{C}	8.34 ± 0.02^{D}

Mean ± standard error (S.E.) values with the same superscripts are not significantly different within the same phase.

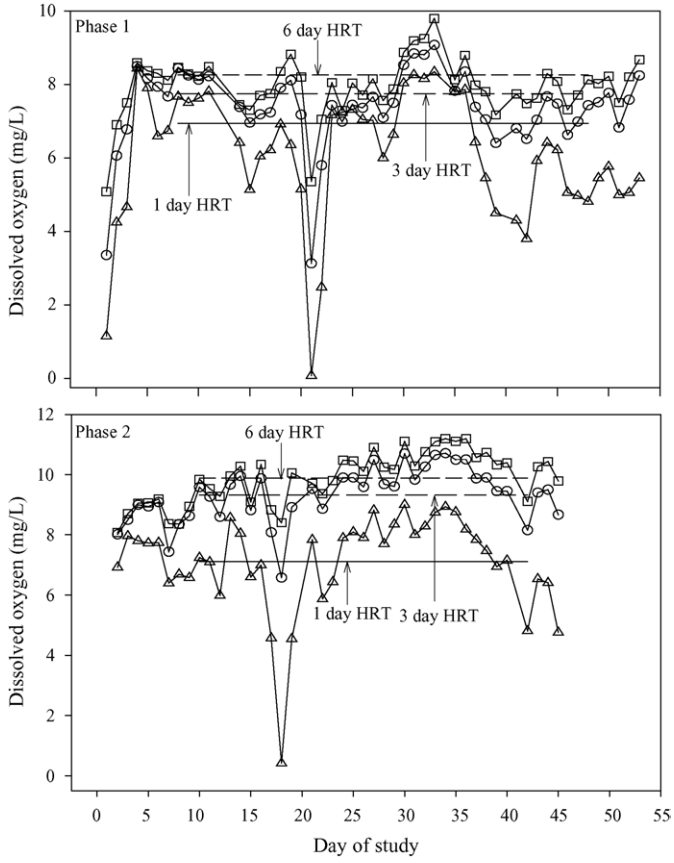

Fig. 2. Mean daily-dissolved oxygen concentration shown for aerated tanks reactors (n = 3 per treatment) operated different hydraulic loadings. Supernatant from gravity thickeners was delivered at flow rates achieving (△) 1-, (○) 3-, and (□) 6-day HRTs. Phase 1 was conducted from July 2002 to August 2002 and Phase 2 was conducted from November 2002 to December 2002.

Table 2
Summary of nitrogen and phosphorous species following aerobic treatment during summer and winter conditions

	Summer phase			Winter phase		
	1-day	3-day	6-day	1-day	3-day	6-day
Dpo_4–P (mg/L)						
Influent		5.97 ± 2.06^{A}			9.93 ± 3.67^{A}	
Effluent	4.98 ± 1.06^{B}	3.43 ± 0.75^{B}C	2.11 ± 0.39^{C}	7.94 ± 1.54^{BC}	7.98 ± 1.57 B	7.40 ± 1.34^{C}
Reduction rate (g/day)	0.49 ± 0.50	0.42 ± 0.30	0.21 ± 0.17	1.48 ± 0.26	0.53 ± 0.14	0.33 ± 0.07
tPO_4–P (mg/L as P)						
Influent		4.98 ± 0.76^{A}			12.93 ± 4.12^{A}	
Effluent	5.69 ± 1.17^{A}	6.83 ± 1.81^{A}	5.41 ± 1.29^{A}	9.97 ± 1.88^{B}	9.77 ± 1.86^{BC}	8.98 ± 1.59^{C}
Reduction rate (g/day)	−0.36 ± 1.96	−0.31 ± 1.14	−0.02 ± 0.38	0.99 ± 0.25	0.32 ± 0.11	0.21 ± 0.06
TAN (mg/L)						
Influent		7.67 ± 1.43^{A}			7.30 ± 1.01^{A}	
Effluent	5.23 ± 0.60^{B}	2.56 ± 0.69^{C}	0.98 ± 0.31-day	4.92 ± 0.32^{B}	3.15 ± 0.55^{C}	3.45 ± 0.87^{C}
Reduction rate (g/day)	1.22 ± 2.26	0.85 ± 0.93	0.55 ± 0.39	1.38 ± 0.23	0.75 ± 0.12	0.35 ± 0.09
NO_2–N (mg/L)						
Influent		0.10 ± 0.04^{A}			0.16 ± 0.07^{A}	
Effluent	1.11 ± 0.17^{B}	2.68 ± 0.56^{C}	2.04 ± 0.43^{BC}	0.08 ± 0.01^{B}	1.61 ± 0.47^{C}	3.10 ± 0.76-day
Production rate (g/day)	0.50 ± 0.44	0.43 ± 0.48	0.16 ± 0.17	−0.05 ± 0.02	0.24 ± 0.08	0.24 ± 0.06
NO_3–N (mg/L)						
Influent		0.36 ± 0.08^{A}			0.74 ± 0.25^{A}	
Effluent	0.73 ± 0.20^{B}	1.55 ± 0.47^{B}	1.28 ± 0.47^{AB}	0.40 ± 0.05^{B}	5.64 ± 1.13^{C}	9.11 ± 1.67D
Production rate (g/day)[1]	0.18 ± 0.46	0.20 ± 0.39	0.08 ± 0.20	−0.19 ± 0.08	0.81 ± 0.17	0.69 ± 0.13

Negative values indicate reduction, Mean ± S.E. values with the same superscripts are not significantly different within the same phase.

Removal of dPO_4–P was consistently achieved through out the study, unlike tPO_4–P removal (Table 2). During the summer period, dissolved phosphorous removal increased to 65% when the HRT was extended to 6 days. Yet during the winter months, dPO_4–P removal was nearly equal, averaging 22%, under all hydraulic regimes. In contrast, total PO_4 removal was only observed during the winter months and was highest at the 6-day HRT (31%).

TAN reduction performance was affected by HRT and water temperature. TAN reduction at the 1-day HRT averaged 32% during both phases, while TAN concentrations were 52–87% lower following 3- and 6-day treatments. As expected, the highest reduction was observed at the 6-day HTR during the warm temperature phase (Table 2). The presence of nitrite and nitrate indicated that nitrification was a factor contributing to TAN reductions. Algal assimilation was also believed to contribute to the TAN reduction; however, this mechanism was not quantified. Stripping has also been identified a factor contributing to the ammonia reduction in aerated lagoons (Oleszkiewicz, 1986; Dumas et al., 1998), but was not likely a significant mechanism during this study. Stripping was considered a minor contributor as ambient pH levels were well below 9.3, the level at which ammonia is easily removed by stripping (Srinath and Loehr, 1974).

Elevated nitrite concentrations suggested that the nitrification process was inhibited. In fully acclimated nitrification systems, NO_2–N levels are generally less than 0.10 mg/L (Metcalf and Eddy, 2003), whereas levels observed during this study were typically greater than 1 mg/L. This was attributed to the organic loading and occasionally potentially limiting oxygen concentrations.

Stenstrom and Song (1991) demonstrated that the nitrifying bacteria are distributed within bio-flocs that also contain heterotrophic bacteria. Because heterotrophic bacteria possess faster growth rates, nitrifiers reside deeper within the floc structure. As a result, oxygen must diffuse deeper into the floc to support nitrification, which means that oxygen levels must be higher in the bulk liquid. Under oxygen-limited

Table 3
Summary of carbon constituents following aerobic treatment during summer and winter conditions

	Summer phase			Winter phase		
	1-day	3-day	6-day	1-day	3-day	6-day
sBOD (mg/L)						
Influent		87.13 ± 24.02^{A}			138.80 ± 18.49^{A}	
Effluent	7.55 ± 2.03^{BC}	7.49 ± 1.97^{B}	8.20 ± 1.84^{C}	27.19 ± 2.65^{B}	23.13 ± 2.74^{C}	22.93 ± 2.95^{C}
Removal rate (g/day)	39.78 ± 20.96	13.27 ± 7.03	6.94 ± 3.96	55.78 ± 5.63	19.26 ± 1.89	9.61 ± 0.96
sCOD (mg/L)[1]						
Influent		168.28 ± 19.22^{A}			219.10 ± 21.75^{A}	
Effluent	40.15 ± 3.19^{BC}	42.10 ± 3.09^{B}	34.67 ± 1.71^{C}	51.14 ± 2.65^{B}	46.79 ± 3.36^{BC}	43.22 ± 2.46^{C}
Removal rate (g/day)	64.06 ± 32.15	21.02 ± 10.36	11.08 ± 4.88	86.69 ± 6.12	29.61 ± 2.09	15.04 ± 1.11
DOC (mg/L)						
Influent		45.70 ± 5.01^{B}			69.39 ± 7.30^{A}	
Effluent	9.77 ± 0.63^{B}	9.62 ± 0.55^{B}	8.22 ± 0.35^{C}	11.00 ± 0.57^{B}	10.68 ± 0.75^{B}	8.96 ± 0.43^{C}
Removal rate (g/day)	17.97 ± 8.26	6.01 ± 2.77	3.11 ± 1.34	29.74 ± 2.12	9.96 ± 0.72	5.10 ± 0.38

Mean ± S.E. values with the same superscripts are not significantly different. Mean influent carbon parameter concentration for each shown in same row as category heading.

conditions, incomplete nitrification results as nitrite oxidation is inhibited more than ammonia oxidation (Metcalf and Eddy, 2003), possibly explaining the nitrite accumulation observed here.

Effluent carbon constituent concentrations were grossly characterized as sBOD, sCOD, and DOC (Table 3). Removal efficiencies for all parameters were relatively constant between the three hydraulic regimes, i.e., the 1-day HRT was as effective as the 3- and 6-day HRTs. Seasonal conditions appeared to have the greatest impact on sBOD removal. The average sBOD removal decreased from 91.1% during to summer to 82.4% during the winter months. Seasonal conditions did not have as much impact on sCOD and DOC removal, which averaged 77.7 and 85.3%, respectively (Table 3).

Aerobic biological treatment of soluble cBOD follows first-order reaction kinetics (Chen et al., 1997). Therefore, in a continuous-flow stirred-tank reactor, operated under steady-state conditions—which is a good approximation of the conditions within these well aerated vessels—the effluent concentration of soluble cBOD (C_e, mg/L) can be predicted from the influent soluble cBOD concentration (C_o, mg/L) at a given mean hydraulic retention time (θ_H, days), when the first-order reaction rate constant (k, 1/day) is known.

$$C_e = \frac{C_o}{1 + k(V/Q)} \quad (1)$$

The mean hydraulic retention time ($\theta_H = V/Q$) was calculated during this research by dividing the water volume (m^3) in the reactor vessel by the wastewater flow rate (m^3/day) through the vessel. As an example, the calculated k-values at the 1-day HRT during our study were determined to be 10.5 and 4.1/day, for the summer (mean $T = 21.3$ °C) and winter (mean $T = 13.5$ °C) conditions, respectively. The calculated k-values where adjusted to 20 °C using the following expression (Metcalf and Eddy, 2003):

$$k_{20} = \frac{k_T}{\theta(T - 20)} \quad (2)$$

Here k_T is the removal constant at ambient temperature (T), and θ is a temperature activity coefficient. During our estimations, we used the typical activity coefficient value of 1.08 for estimating k_{20} values of the seasonal means previously stated, based on the range of values provided for aerated lagoons (Metcalf and Eddy, 2003). Thus, calculated k_{20} values for soluble cBOD during the summer and winter months were 9.5 and 6.8/day, respectively. These values were similar to removal rates presented by Brenes et al. (2000) for propionic acid.

The observed similarity between the rate constants determine during this study and that determined from fatty acids by Brenes et al. (2000) tends to provide some confirmation our assertion that hydrolosis

followed by fermentation occurred during solids storage in the gravity thickener. van Rijn et al. (1995) demonstrated that the organic constituents released following fermentation of an aquaculture sludge were readily degraded by facultative bacteria. Subsequent work by Aboutboul et al. (1995) identified much of this degradable substance as volatile fatty acids (acetate, propionate, and butyrate) that were extremely labile to microbial decomposition. Based on the soluble cBOD treatment efficiencies measured during the present study, the fresh supernatant discharged from the gravity thickening tank contained approximately 10–20% of more recalcitrant compounds.

Chapter 10 of the Soil Conservation Service's (SSC, 1992) Agricultural Waste Management Field Handbook provides design specifications for sizing non-mechanically aerated aerobic lagoons that are based upon a total BOD_5 loading rate and a water depth of 0.6–1.5 m (2–5 ft). In a climate such as is found at the Freshwater Institute, the design loading rate would be 15–17 kg (34–37 lb) total BOD_5/acre, which would produce a passive aerobic lagoon with a mean hydraulic retention time of approximately 40–50 days. The passive non-aerated aerobic lagoon provides a much lower rate of treatment, and thus requires a much longer treatment interval than would be required by the actively aerated treatment vessels described in this research.

In an actively aerated vessel, the oxygen demand is met using surface agitators or air diffusers (more common in aerated basins). To achieve the desired cBOD removal, sufficient oxygen must be supplied by the aeration equipment, which can be sized from oxygen demand estimates using:

$$\text{Lbs. of } O_2/\text{day} = \frac{Q \times S_r \times 8.34}{0.68} \quad (3)$$

where Q (Mgal/day) is the influent flow rate, S_r (mg/L) is the amount of soluble substrate removed, and 8.34 and 0.68 are conversion factors (Metcalf and Eddy, 2003).

Using the 1-day HRT cBOC removal rate of 79.5 mg/L, treating the entire Freshwater Institute supernatant stream (Q = 3.5 gal/min) requires approximately 4.9 lbs O_2/day (2.2 kg O_2/day). A treatment basin of 19 m^3 is required for the 1-day HRT. The power requirement to achieve this level of treatment using surface agitation can be determined following Metcalf and Eddy (2003):

$$P = \frac{\text{kg of } O_2/\text{day}}{N} \quad (4)$$

where N is the transfer rate (kg O_2/kW h) for the selected aerator corrected for local conditions (i.e. maximum temperature and altitude) using:

$$N = N_o \left(\frac{\beta C_{Walt} - C_L}{C_{s20}} \right) 1.024^{T-20} \alpha \quad (5)$$

Here N_o is the oxygen transfer rate in water at 20 °C (obtained from manufacturer, usually with a initial dissolved oxygen concentration = 0), β is the salinity–surface tension correction factor (usually 1), C_{Walt} (mg/L) is the oxygen saturation concentration for tap water at given temperature and altitude, C_{S20} (mg/L) is the oxygen saturation concentration in tap water at 20 °C, C_L (mg/L) is the desired ambient oxygen concentration, T (°C) is the field temperature, and α is the oxygen-transfer correction factor for waste water (ranges between 0.80 and 0.95 for domestic waste water and assumed to be 85 for this example). Using the summer conditions under which this study was conducted, C_{Walt} = 7.7, C_L = 2, C_{S20} = 8.9, T = 28 °C (maximum observed), α = 0.9, and N_o = 1.8 kg O_2/kW h (Kasoc® surface aerator, model F3400-A), the power requirement would be 1.2 kW.

During this study, aerobic treatment occurred as a result of microbiological degradation and conversion of the organic carbon, nitrogen and phosphorus compounds in the supernatant. TSS concentrations increased significantly during aerobic treatment at all HRTs under both summer and winter conditions (Table 4). Suspended solids concentration increased an average 64% during the summer period and 20% during the winter period. Tanks were clearly green with algae during both summer and winter conditions. However, photosynthetic activity was not quantified during either study phase. TSS measurements were similar across the hydraulic treatments during summer conditions, averaging 164 mg/L. Yet, during the winter phase, TSS concentration was significantly higher at the 1-day HRT, which also had the highest substrate loading rate. The ratio of volatile and total

Table 4
Summary of effluent TSS and VSS following aerobic treatment during summer and winter conditions

	Summer phase			Winter phase		
	1-day	3-day	6-day	1-day	3-day	6-day
TSS (mg/L)						
Influent		100.12 ± 17.73^{A}			101.60 ± 7.69^{A}	
Effluent	167.23 ± 12.32^{B}	160.49 ± 19.37^{B}	166.18 ± 14.28^{B}	141.70 ± 6.48^{B}	112.00 ± 8.08^{A}	113.10 ± 8.37^{A}
Removal rate (g/day)	-33.55 ± 40.42	-10.06 ± 19.90	-5.48 ± 7.88	-21.40 ± 2.36	-2.19 ± 1.24	-1.18 ± 0.71
VSS (mg/L)						
Influent		87.52 ± 15.90^{A}			88.91 ± 6.65^{A}	
Effluent	146.26 ± 10.83^{B}	136.77 ± 16.46^{B}	138.67 ± 11.85^{B}	122.90 ± 5.20^{B}	95.36 ± 6.69^{A}	96.13 ± 6.79^{A}
Removal rate (g/day)	-29.37 ± 35.09	-8.21 ± 16.98	-4.24 ± 6.57	-21.40 ± 2.36	-1.48 ± 1.05	-0.80 ± 0.57
VSS/TSS ratio	87.42 ± 0.83^{A}	86.54 ± 0.96^{AB}	84.34 ± 0.94^{B}	87.10 ± 0.44	85.42 ± 0.63	85.59 ± 0.68

A negative removal rate indicates a net increase in parameter concentration following treatment. Mean ± S.E. values with the same superscripts are not significantly different.

suspended solids remained constant across HRTs and environmental conditions, averaging 87.2%.

4. Conclusions

Replicated experimental tank studies on aerobic treatment at 1-, 3-, and 6-day HRT showed that approximately 90% of soluble BOD, 80% of dissolved organic carbon, and 75–80% of soluble COD could be removed from the thickening tank overflow and was relatively independent of HRT. TAN removal, however, was strongly influenced by HRT with TAN removal increasing from 32 to 87% from the 1-day HRT treatment as compared to the 6-day HRT treatment. TAN reduction principally resulted from biological transformation via the nitrification pathways and algal or microbial assimilation. With respect

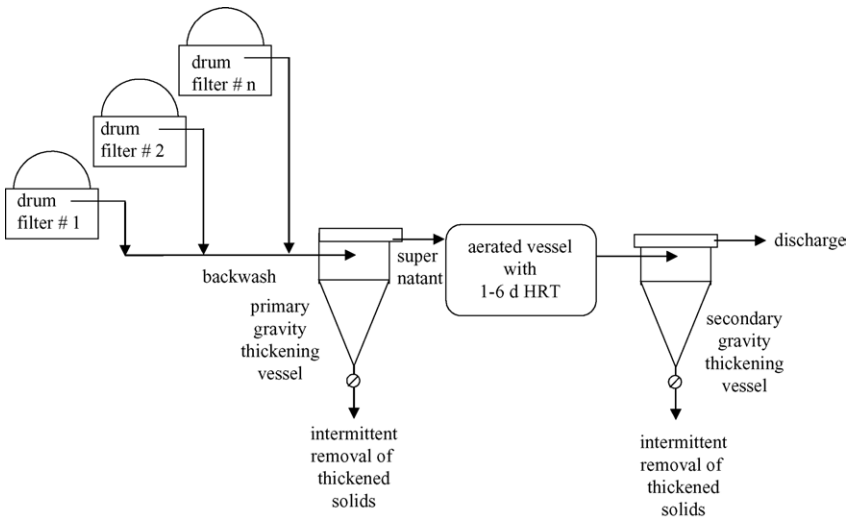

Fig. 3. This process flow drawing provides an example of how to treat the solids-laden microscreen filter backwash from an intensive fish farm. Solids would be concentrated in a gravity thickening tank and removed from the base of the tank on a weekly or monthly basis, as necessary. Supernatant discharged from the gravity thickening tank would be treated within an aerated basin that provides at least a 1-day mean hydraulic retention time (or a 6-day HRT if lower TAN concentrations were desired). Water exiting the aerated basin would pass through a second radial-flow settler before being discharged.

to the transformation process, nitrogen in the bulk solution would be conserved, and while ammonia concentrations would be lowered, the total nitrogen concentration was not reduced. This is particularly important as effluent quality permits are increasingly moving toward total nitrogen limits to reduce to the pollution potential of aquaculture effluents. Thus, effluent streams have to be processed to achieve nitrogen reduction, not just transformation. Total nitrogen reductions could be achieved by subjecting the effluent from the aerobic system to settling (Fig. 3) or to anoxic conditions to facilitate denitrification. This treatment strategy could be achieved using a sequencing batch reactor, where aqueous conditions alternate between aerobic (promotion of nitrification) and anoxic (promotion of denitrification), thereby obtaining the desired nitrogen reduction for pollution abatement. The effect of aerobic treatment on phosphorus removal was less clear. An aerobic treatment system as described here would have to be followed by a solids capture process (e.g., radial-flow settling tank) in order to capture the 30–170 mg/L of TSS overflowing the aerobic treatment vessel (Fig. 3).

Acknowledgments

This material is based upon work supported by the Agricultural Research Service, United States Department of Agriculture, under grant agreement numbers 59-1930-1-130 and 59-1930-8-038. We thank James Ebeling, Carla Welsh, Sarah Reaves, Susan Glenn, and Kata Rishel for their assistance with this research.

References

Aboutboul, Y., Arviv, R., van Rijn, J., 1995. Anaerobic treatment of intensive fish culture effluents: volatile fatty acid mediated denitrification. Aquaculture 133, 21–32.

Adler, P.R., Sikora, L.J., 2005. Mesophilic composting of Arctic char manure. Comp. Sci. Utilization 13, 34–42.

Adler, P.R., Harper, J.K., Takeda, F., Wade, E.M., Summerfelt, S.T., 2000. Economic evaluation of hydroponics and other treatment options for phosphorus removal in aquaculture effluent. Horticult. Sci. 35, 993–999.

Adler, P.R., Takeda, F., Glenn, D.M., Wade, E.M., Summerfelt, S.T., Harper, J.K., 1996. Phytoremediation of wastewater using lettuce and basil. Proc. Am. Soc. Plasticulturists 26, 253–258.

APHA, 1998. Standard Methods for the Examination of Water and Wastewater, 20th ed. American Public Health Association, Washington, DC.

Bastian, R.K., 1992. Overview of federal regulations pertaining to aquaculture waste management and effluent. In: Blake, J., Donald, J., Magette, W. (Eds.), National Livestock, Poultry, and Aquaculture Waste Management. American Society of Agricultural Engineers, St. Joseph, Michigan, pp. 220–226.

Bergheim, A., Brinker, A., 2003. Effluent treatment for flow through system and European environmental regulations. Aquacult. Eng. 27, 61–77.

Bergheim, A., Cripps, S.J., Liltved, H., 1998. A system for the treatment of sludge from land-based fish-farms. Aqua. Living Resour. 11, 279–287.

Brenes, M., Garcia, P., Romero, C., Garrido, A., 2000. Treatment of green table olive waste waters by an activated-sludge process. J. Chem. Technol. Biotechnol. 75, 459–463.

Chen, S., Coffin, D.E., Malone, R.F., 1997. Sludge production and management for recirculating aquaculture systems. J. World Aquacult. Soc. 28, 303–315.

Chen, S., Summerfelt, S.T., Losordo, T., Malone, R., 2002. Recirculating systems, effluents, and treatments. In: Tomasso, J.R. (Ed.), Aquaculture and the Environment in the United States. Aquaculture Society, A Chapter of the World Aquaculture Society, Baton Rouge, LA, U.S., pp. 119–140.

Cripps, S.J., Bergheim, A., 2000. Solids management and removal for intensive land-based aquaculture production systems. Aquacult. Eng. 22, 33–56.

Cripps, S.J., Kelly, L.A., 1996. Reductions in wastes from aquaculture. In: Baird, D.J., Beveridge, M.C.M., Kelly, L.A., Muir, J.F. (Eds.), Aquaculture and Water Resource Management. Blackwell Science, Inc., Cambridge, MA, pp. 166–201.

Davidson, J., Summerfelt, S.T., 2005. Solids removal from a cold-water recirculating system—comparison of a swirl separator and a radial-flow settler. Aquacult. Eng. 33, 47–61.

Dumas, A., Laliberte, G., Lessard, P., de la Noue, J., 1998. Biotreatment of fish farm effluents using the cyanobacterium *Phormidium bohneri*. Aquacult. Eng. 17, 57–68.

Ebeling, J.M., Sibrell, P.L., Ogden, S., Summerfelt, S.T., 2003. Evaluation of chemical coagulation-flocculation aids for the removal of phosphorus from recirculating aquaculture effluent. Aquacult. Eng. 29, 23–42.

Ebeling, J.M., Ogden, S., Sibrell, P.L., Rishel, K.L., 2004. Application of chemical coagulation aids for the removal of suspended solids (TSS) and phosphorus from the microscreen effluent discharge of an intensive recirculating aquaculture system. J. North Am. Aquacult. 66, 198–207.

EPA (Environmental Protection Agency), 2004. Effluent Limitations Guidelines and New Source Performance Standards for the Concentrated Aquatic Animal Production Point Source Category; Final Rule. 40 CFR part 451. Washington, DC.

Hosetti, B.B., Frost, S., 1995. A review of sustainable value of effluents and sludges from wastewater stabilization ponds. Ecol. Eng. 5, 421–431.

IDEQ (Idaho Division of Environmental Quality), 1998. Idaho Waste Management Guidelines for Aquaculture Operations, Idaho Department of Health and Welfare, Division of Environmental Quality, Twin Falls, Idaho.

Metcalf and Eddy, Inc., 2003. Wastewater Engineering, Treatment and Reuse, fourth ed., McGraw-Hill, New York.

Michael, J.H., 2003. Nutrients in salmon hatchery wastewater and its removal through the use of a wetland constructed to treat off-line settle pond effluent. J. World Aquacult. Soc. 226, 213–225.

Oleszkiewicz, J.A., 1986. Nitrogen transformations in an aerated lagoon treating piggery wastes. Agricult. Wastes 16, 171–181.

Porrello, S., Lenzi, M., Persia, E., Tomassetti, P., Finoia, M.G., 2003a. Reduction of aquaculture wastewater eutrophication by phytotreatment pond systems I Dissolved and particulate nitrogen and phosphorus. Aquaculture 219, 515–529.

Porrello, S., Lenzi, M., Persia, E., Tomassetti, P., Finoia, M.G., Mercatali, I., 2003b. Reduction of aquaculture wastewater eutrophication by phytotreatment pond systems II Nitrogen and phosphorus content in macroalgae and sediment. Aquaculture 219, 531–544.

Srinath, E.G., Loehr, R.C., 1974. Ammonia desorption by diffused aeration. J. Water Pollut. Control Feder. 46, 1939–1957.

SCS (Soil Conservation Service), 1992. Agricultural Waste Management Field Handbook. United States Department of Agriculture, Soil Conservation Service, Washington, DC.

Stenstrom, M.K., Song, S.S., 1991. Effects of oxygen transport limitation on nitrification in the activated-sludge process. Res. J. Water Pollut. Control Feder. 63, 208–219.

Summerfelt, S.T., 1999. Waste handling systems. In: Bartali, E.H., Wheaton, F.W. (Eds.), CIGR Handbook of Agricultural Engineering, Volume II: Animal Production and Aquacultural Engineering. American Society of Agricultural Engineers, St. Joseph, MI, pp. 309–350.

Summerfelt, S.T., Adler, P.R., Glenn, D.M., Kretschmann, R., 1999. Aquaculture sludge removal and stabilization within created wetlands. Aquacult. Eng. 19 (2), 81–92.

Summerfelt, S.T., Vinci, B.J., 2003. Best Waste Management Practices for Recirculating Systems. In: Summerfelt, R.C., Clayton, R.D. (Eds.), Aquaculture Effluents: Overview of EPA Guidelines and Standards and BMPs for Ponds, Raceways, and Recycle Systems. Publication Office, North Central Regional Aquaculture Center, Iowa State University, Ames, Iowa, pp. 111–133.

van Rijn, J., Fonarev, N., Berkowitz, B., 1995. Anaerobic treatment of intensive fish culture effluent: digestion of fish feed and release of volatile fatty acids. Aquaculture 133, 9–20.